Chapter 9

The Impact of Groundwater Flows on Estuaries

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Introduction

Direct groundwater discharge to the coastal ocean occurs as submarine seeps and occasionally springs. These discharges occur most frequently along or near the shoreline but are also known to occur quite far offshore on the continental shelf (Moore, 1999; Karpen and others, 2004). These discharges (Figure 9-1) are largely unseen and difficult to quantify because their specific discharge rates are typically low (for example, 1–27 cm per day) (Burnett and others, 2003). Consequently this submarine component of the hydrologic cycle has received historically little attention. However, recent concerns about coastal water quality have prompted increased interest in characterizing and quantifying this submarine groundwater discharge (SGD) and its biogeochemical implications (Moore, 1999). Numerous studies based on direct and indirect measurements have now provided evidence that the exchange of water between coastal sediments and surface waters can be a substantial fraction of surface freshwater inflow (for example, Sewell, 1982; Cable and others, 1996; Moore, 1997; Charette and others, 2001). Because the seepage areas involved are large, the total discharge can be high even where the specific discharge rates are low (Burnett and others, 2003). The chemical implications are even more significant because groundwater is typically enriched relative to surface water in many dissolved constituents such as nutrients and metals. Though our understanding of these processes is still incomplete, there is already evidence that these chemical and water fluxes may have important ecological consequences. The following is a review of our current state of understanding of the scope and processes involved in SGD; the methods used to investigate SGD, their current limitations, and some of the ways researchers are trying to overcome them; and the chemical and ecological consequences of SGD on estuaries.

Submarine Groundwater Discharge

SGD refers to the mixture of terrestrial advecting groundwater and saline recirculated seawater that discharges directly to the coastal ocean (Moore, 1999). Estimates of the terrestrial advecting fraction of SGD are between six and ten percent of surface water inputs; however, total SGD discharge can be much greater due to the recirculated seawater component (Burnett and others, 2003). While municipal water managers have historically been interested in the discharge of

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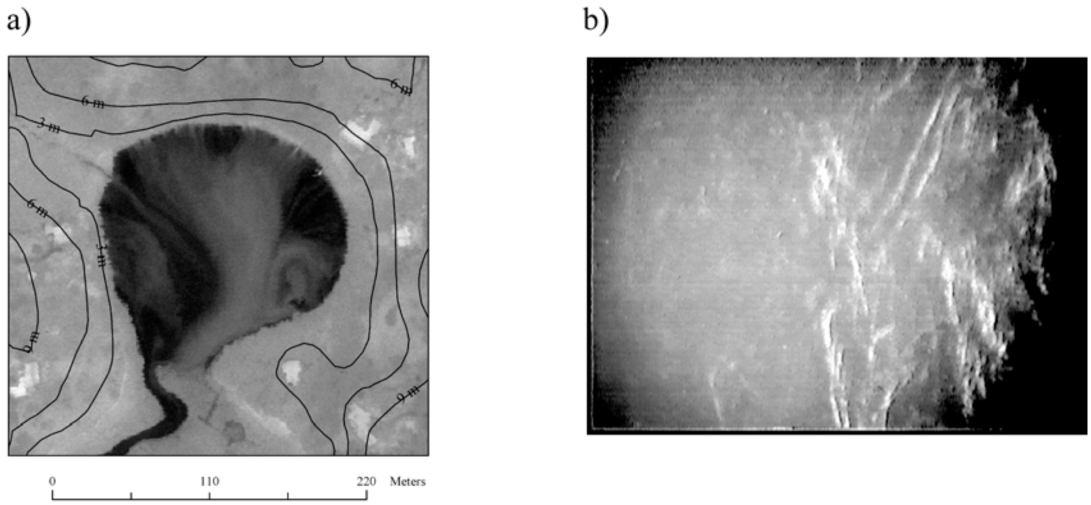

Figure 9-1. Though in many ways difficult to observe, submarine groundwater discharge can be imaged by a) infrared photography (image provided by Ann Mulligan and Matt Charette) and b) 'schlieren' photographic techniques sensitive to the density differences between saline and fresh or brackish groundwater (reprinted from Karpen and others, 2004, copyright 2003, with permission from Elsevier). In this a) thermal infrared aerial photograph of a tidal pond near Waquoit Bay, MA, the cooler (lighter) groundwater discharge can be seen mixing with the warmer (darker) bay water (Mulligan and Charette 2005). In b) a specially designed underwater 'schlieren' photographic system images submarine seepage several kilometers offshore in the Baltic Sea (Karpen and others, 2004).

terrestrial groundwater to the ocean, the total SGD must be quantified when considering the biogeochemical effects on estuaries.

A variety of driving forces are involved in controlling the discharge of this terrestrial and seawater mixture (see Burnett and others, 2003 for a detailed review). The terrestrial hydraulic gradient and the permeability and thickness of aquifer materials and bay bottom sediments control the discharge and spatial heterogeneity of both terrestrial and entrained recirculated seawater. Tidal and wind driven changes in sea level influence the hydraulic gradient as well as induce wave and tidal pumping of bay bottom and shoreline sediments (Burnett and others, 2003). Density differences between fresh groundwater and more saline or even hypersaline surface waters in salt marshes and tidal flats can result in buoyant forces and density driven convection (Simmons and others, 1991). A consequence of this complex mixture of fluids and driving forces is that SGD varies temporally and spatially even at relatively small scales, making it difficult to locate areas of significant submarine discharge difficult and relate the results of large and small scale SGD investigations (Breier and others, in review).

Quantifying Submarine Groundwater Discharge

There are three basic approaches to quantifying SGD including hydrogeologic modeling, direct seepage measurements, and chemical tracer mixing models (Burnett and Dulaiova, 2003; Oberdorfer, 2003). However, these methods do not all measure the same components of SGD and are not necessarily directly comparable. Hydrogeologic modeling has typically been used to estimate the advecting terrestrial component of SGD. More recently, variable density groundwater transport algorithms have been used to include the recirculated seawater component (Oberdorfer, 2003). Seepage meters are used in relatively small areas and are especially useful in areas where SGD is already known to occur. Seepage meter measurements reflect total SGD; if discharge salinity is also measured, then the advecting groundwater and recirculated seawater components can be also be estimated. Chemical tracers are the most common approach because they can be used to estimate SGD to large areas such as entire bays or regions. Chemical tracer estimates generally reflect some combination of advecting groundwater and recirculated seawater discharge, depending on the specific tracer.

Estimates of total SGD are theoretically possible if the nearshore hydraulic gradient and sediment permeability are well known. In practice, obtaining the necessary data in other than the first few meters of sediment requires extensive drilling, geophysical surveying, or extrapolation of sparse data. Langevin (2001) used the U.S. Geological Survey's SEAWAT variable density groundwater transport code to model total SGD to Biscayne Bay, Florida, with mixed results. Both regional three-dimensional and local-scale two-dimensional vertical models were developed assuming steady-state conditions and homogeneous aquifer characteristics. While the variable density transport code was capable of estimating both the advecting groundwater and recirculated seawater components of SGD, it was not able to accurately simulate the groundwater salinities beneath Biscayne Bay as observed in monitoring wells. Additionally, surface aquifer recharge rate and the terrestrial boundary groundwater flux must be estimated and the aquifer hydraulic parameters adjusted to calibrate the modeled water-table elevations to monitoring well levels (Langevin, 2001). Similar boundary conditions and calibrations are necessary in most if not all hydrogeologic models and introduce considerable uncertainty in their results.

SGD can be directly measured at the sediment/water interface using a seepage meter (for example, Cable and others, 1997; Michael and others, 2003) which collects seepage through a small area of the sediment surface. In addition to being labor intensive, seepage meters have an inherent bias which underestimates discharge due to the increased hydraulic friction associated with the meter (Cable and others, 1997). Further, because SGD is frequently heterogeneous, seepage meter results exhibit large variability (Michael and others, 2003) and only limited extrapolation of results to larger areas is possible.

Chemical tracers such as radium, radon, and methane provide an integrated spatial signal allowing quantification of SGD throughout entire bay systems (for example, Rama and Moore, 1996; Krest and others, 1999; Charette and others, 2003; Breier and others, 2004). The ideal chemical tracer of SGD is a dissolved constituent which (1) exhibits a substantial enrichment in groundwater relative to other potential end-member waters (for example, seawater, river water, rain, and runoff) and (2) behaves conservatively within the coastal zone (Charette and others, 2001). Radon is perhaps closest to the ideal; it is highly enriched in groundwater, as a noble gas exhibits very conservative behavior, and is relatively straightforward to measure (Cable and others, 1996; Burnet and Dulaiova, 2003). Radium isotopes are also powerful tracers of SGD because they behave conservatively in brackish and marine waters and are enriched in groundwater (Krest and others, 1999). They also provide a means of estimating bay residence time and tidal transport which is essential to properly modeling tracer mixing within a study area (Charette and others, 2001). Methane is a product of anaerobic decay and is found in high concentrations in anoxic groundwaters with sufficient organic matter for methanogenesis but is subject to microbial uptake and production and is not strictly conservative (Bugna and others, 1996). Regardless of the trace, SGD is estimated from a mixing model for the chemical species in question (for example, Breier and Edmonds, in review).

While the techniques just mentioned can provide valuable data, they can also involve substantial uncertainty. In particular, determining the spatial distribution of SGD in a study area can be very challenging. While natural chemical tracers are useful at estimating total discharge to an area, they cannot be used to pinpoint the source of discharge because water column mixing weakens and spatially integrates the signal. Conversely, while direct measurements with seepage meters can be used to measure discharge at a point, they do not capture spatial variation in the system and can miss significant localized discharges altogether. In fact, only in a few well studied areas is the spatial distribution of SGD understood at a scale approaching that at which organisms experience its effects. This has hampered attempts at studying the ecological consequences of SGD. Recently, the use of simultaneous geochemical and geophysical surveying (Figure 9-2) and thermal infrared aerial photography have demonstrated that there are ways to acquire more detailed spatial data on SGD (Bratton and others, 2004; Breier and others, in review; Mulligan and Charette, 2005). Data from techniques will help plan future field studies which more directly test the biogeochemical and ecological hypotheses concerning SGD.

Chemical and Ecological Consequences of Submarine Groundwater Discharge

Johannes (1980) was one of the first to seriously discuss groundwater as a potential pathway for nutrients to coastal estuaries. Because groundwater is often enriched in natural and anthropogenic nutrients, SGD may be ecologically important even where discharge rates are small compared to surface water inputs. If SGD does represent an important control on estuarine salinity and chemical cycling, particularly nutrients, then it is reasonable to suspect that SGD dynamics and distribution may also affect ecosystem processes (Johannes, 1980). Two widely expressed concerns are that (1) fluctuations in SGD rates are related to the initiation of nuisance algal blooms (Sewell, 1982; Laroche and others, 1997) and (2) anthropogenic increases in groundwater nutrient concentrations are partially responsible for the increasing eutrophication of coastal waters (for example, Johannes, 1980; Laroche and others, 1997).

Perhaps the most widely discussed hypothesis related to SGD is that changes in the associated nutrient flux may initiate algal blooms. Evidence supporting this comes from a study by Laroche and others (1997) of 11 years of well levels, coastal salinities, nutrient concentrations, and cell counts of the brown tide species *Aureococcus anophagefferens* in Peconic Bay, Long Island. Laroche and others (1997) showed that bloom intensity was inversely proportional to well levels and directly proportional to bay salinities. However, bay salinity does not itself appear to be the direct initiator of *A. anophagefferens* blooms. The salinity of Peconic Bay during the study years was within the optimal growth range of *A. anophagefferens* growth 98 percent of the time. Instead, high bay salinity is a result of low SGD, and when SGD is low, the ratio of organic to inorganic nitrogen in the bay increases. *A. anophagefferens* is believed to have a competitive advantage in utilizing organic nitrogen compared to other algal species and increased inorganic nitrogen has been shown to inhibit *A. anophagefferens* growth. Local groundwater in the Peconic

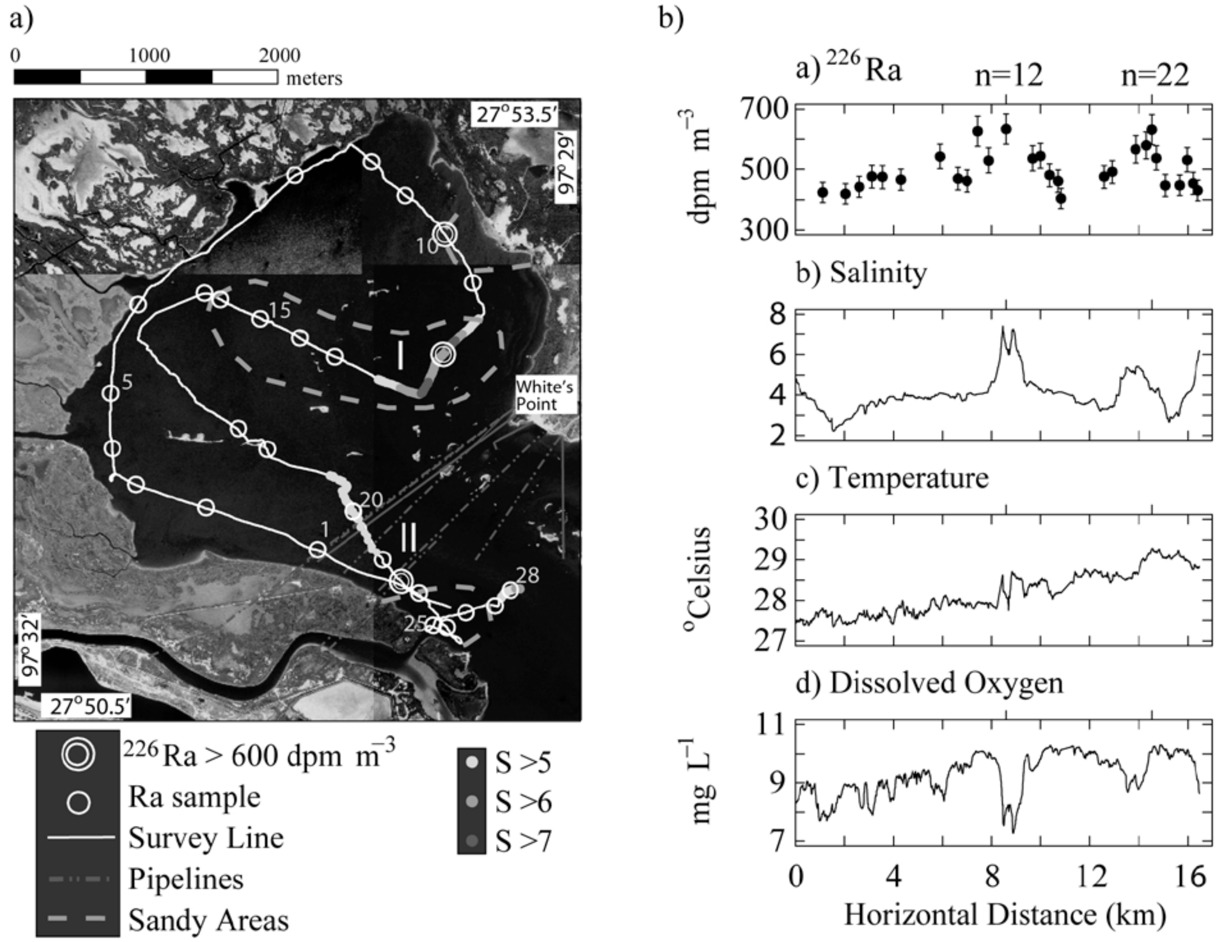

Figure 9-2. A synoptic geophysical and geochemical survey was used to investigate SGD to upper Nueces Bay, Texas (Breier, in review). The survey incorporated continuous resistivity profiling; measurements of surface water salinity, temperature, and dissolved oxygen; and point measurements of dissolved radium isotopes. The survey revealed areas of interleaving, vertical fingers of high and low conductivity extending up through 7 m of bay bottom sediments into the surface water, located within 100 m of surface salinity and dissolved radium maxima along with peaks in water temperature and lows in dissolved oxygen. These results indicate either brackish submarine groundwater discharge or the leakage of oil field brine from submerged petroleum pipelines. Survey with radium samples marked by white circles; highest activity radium samples are marked by concentric circles. Areas of high surface salinity are light to dark grey.

Bay is high in inorganic nitrogen, so in wet years when the water table and SGD are high more inorganic nitrogen is delivered to the bay. In dry years when SGD is low recycled organic nitrogen from bay bottom sediments becomes more important. In addition, the seasonal timing of bloom initiation and peak corresponds with the seasonal decline in water table height due to higher summertime evapotranspiration and decreased groundwater recharge. This study of Peconic Bay required a lengthy environmental time series and has yet to be completed in other areas, but it provides a valuable example of how declines in groundwater discharge can impact bay ecology.

Increases in SGD and/or the associated nitrogen flux can also impact bay and coastal ecology. Increases in SGD-delivered inorganic nitrogen can stimulate aquatic plant growth. Maier and Pregnall (1990) showed that the growth of the eelgrass *Zostera Marina* and the macroalgae S*argassum filipendula* and *Enteromorpha intestinalis* are all stimulated by nitrate enrichment. Typically eelgrass utilizes ammonium for growth; however, in sandy beaches around Woods Hole, Massachusetts, eelgrass growing in areas of SGD were shown to have elevated nitrate reductase enzyme activity, indicating that they were utilizing SGD-derived nitrate for growth. The macroalgal species growing in the same area exhibited even higher nitrate reductase activity, indicating that they are more successful at utilizing nitrate for growth. Maier and Pregnall (1990) suggest that at certain levels groundwater nitrate flux may influence and stimulate the growth of marine plants like the eelgrass *Z. Marina*, but when the groundwater nitrate flux is higher the growth of macroalgae takes over and can ultimately smother the bottom plants. A similar process has been observed in Jamaican and Floridian coral reefs where an increase in dissolved inorganic nitrogen in near bottom waters has stimulated the growth of epilithic macroalgae at the expense of the corals on which they grow (Lapointe, 1997). This process has been going on for some time and in the case of the Jamaican coral reefs the macroalgae now dominates. The elevated nitrate and low salinity in near bottom waters of these reefs indicates that the nitrate is being delivered by groundwater. In addition, analysis of macroalgae tissue shows elevated $d^{15}N$ which suggests that the nitrate is ultimately coming from groundwater contaminated by wastewater (Lapointe, 1997).

SGD may also influence estuarine ecology in ways other than modifying nutrient supply. First, the spatial distribution of SGD within an embayment may create environmental refugia for some organisms (for example, Nielsen and Lisle, 1994). Freshwater fish such as steelhead trout use pools influenced by groundwater discharge as thermal refuge (Nielsen and Lisle, 1994). Aquatic biota may also use such areas to escape chronic hypoxia particularly in times of drought or low river discharge (Magoulick and Kobza, 2003). Second, since the dissolved O_2 concentration of groundwater varies, suboxic or anoxic SGD may be a mechanism for the remobilization of redox sensitive transition metals in bay sediments (Liu and others, 2001).

Conclusions

Evidence of significant submarine groundwater fluxes has been growing in recent years. SGD investigations to date have largely focused on locating and quantifying the amounts of SGD to different areas. Limited but provocative evidence of the ecological effects of these discharges on harmful algal bloom initiation and eutrophication have already been found. Other ecological impacts are suspected but remain to be tested. Methods and techniques are just now reaching the point that we can identify and investigate these fluxes on the very small scales at which organisms experience them.

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